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Assessing resource depletion in LCA: a review of methods and methodological issues

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Abstract
Purpose Political interest in the future availability of natural resources has spiked recently, with new documents from the European Union, United Nations Environment Programme and the US National Research Council assessing the supply situation of key raw materials. As resource efficiency is considered a key element for sustainable development, suitable methods to address sustainability of resource use are increasingly needed. Life cycle thinking and assessment may play a principal role here. Nonetheless, the extent to which current life cycle impact assessment methods are capable to answer to resource sustainability challenges is widely debated. The aim of this paper is to present key elements of the ongoing discussion, contributing to the future development of more robust and comprehensive methods for evaluating resources in the life cycle assessment (LCA) context.
Methods We systematically review current impact assessment methods dealing with resources, identifying areas of improvement. Three key issues for sustainability assessment of resources are examined: renewability, recyclability and criticality; this is complemented by a cross-comparison of methodological features and completeness of resource coverage.
Results and discussion The approach of LCA to resource depletion is characterised by a lack of consensus on methodology and on the relative ranking of resource depletion impacts as can be seen from a comparison of characterisation factors. The examined models yield vastly different characterisations of the impacts from resource depletion and show gaps in the number and types of resources covered.
Conclusions Key areas of improvement are identified and discussed. Firstly, biotic resources and their renewal rates have so far received relatively little regard within LCA; secondly, the debate on critical raw materials and the opportunity of introducing criticality within LCA is controversial and requires further effort for a conciliating vision and indicators. We identify points where current methods can be expanded to accommodate these issues and cover a wider range of natural resources.

Keywords Critical resources · LCIA methodology · Life cycle impact assessment · Resource depletion

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1 Introduction

The survival of the global economy at present and increasing output levels depends on its sustained supply with natural resources. The depletion of abiotic and biotic resources is therefore a fundamental issue for sustainability assessment, entailing and affecting environmental and socioeconomic. As the UN Environmental Programme's International Panel on Sustainable Resource Management points out (UNEP 2010), permanent depletion of non-renewable fossil fuels and use of high-grade metal ores can pose a threat to sustained industrial production, just as the harvest of living resources such as wood and fish above renewal rates threatens their reproduction and future availability. Availability for human use is not equally critical for all resources (cf. EC 2010a; NRC 2007; UNEP 2010); moreover, the threat to supply safety for individual resources is

likely to shift over time, as will policy priorities. Periodical assessment of the supply of resources and a move towards higher recycling rates are among the main objectives of the European Union's flagship initiative on a resource-efficient Europe (EC 2011a) and the European Raw Materials Initiative (EC 2008), which in particular calls for the identification and monitoring of critical raw materials to ensure a safe supply in the future. Resource depletion, as an impact category, has a very immediate economic and geopolitical aspect (i.e. continued supply), apart from impacts on the environment and human health. In a sustainability assessment of natural resources, these perspectives should be considered as well as modelled and assessed, in order to support decision making regarding environmental as well as said socioeconomic concerns.

In the last 30 years, a number of methodologies and indicators for resource depletion have been developed, some including economic aspects related to abiotic and biotic resource consumption (cf. a recent review of Giljum et al 2011). Material-flow-based indicators such as Domestic Material Consumption, Environmentally Weighted Material Consumption and National Accounts Matrix Extended by Environmental Accounts seek to quantify environmental pressures from resource consumption in national accounting. The Ecological Footprint, Human Appropriation of Net Primary Production and Land and Ecosystem Accounts pursue a similar goal with a focus on biotic resources (cf. van der Voet et al. 2009; European Environment Agency) 2010).

In the context of assessing resource efficiency, there is the tendency of adopting indicators simply based on mass aggregation. For example, at EU level, an indicator given as the ratio of GDP to domestic material consumption—expressed in Euro/tonne—has been adopted (Eurostat 2013). A higher ratio would indicate better performance, with growth consuming relatively fewer resources. The limitation of the indicator is that it captures only the material resource aspects and does not deal with other resources (EU 2011a) or the potential shift of burdens across countries.

Life cycle assessment (LCA) methodology—due to its systemic approach—is considered apt for comprehensive environmental assessment and suitable to provide a valuable support in integrating sustainability of resources into design, innovation and evaluation of products and services (Sala et al. 2012). Resources could be evaluated in relation to their depletion (consumption related to geological/ natural reserve), scarcity (economic availability of a resource) and their criticality (a resource that is scarce and also crucial for society).

A considerable range of methodologies for assessing resource depletion in LCA has been proposed, with different theoretical underpinnings. However, it is arguable whether resource availability is an environmental or economic issue and whether this should be subject to characterisation models. For example, Weidema et al (2005) suggest a sharper distinction between human activity and the natural environment. They argue that future impacts from resource extraction, and future (backup) extraction technologies, ought to be defined in the life cycle inventory as they pertain to human activities; the biophysical impacts of resource use to be remedied—e.g. diminishing ore stocks—are to be recorded in the life cycle impact assessment (LCIA) as they occur in the environment. Even so, they acknowledge the unresolved issue of socioeconomic impacts, as impacts on human activity. In this regard, the ecologically based LCA (Eco-LCA) approach put forward by Zhang et al. (2010) proposes a hierarchy or resource use indicators aiming at capturing the damage to ecosystem services—the functions these resources fulfil—stemming from resource use.

Currently, LCIA methods are based upon different definitions of the depletion problem (Steen 2006), which could be summarised as: (1) assuming that mining cost will be a limiting factor, (2) assuming that collecting metals or other substances from low-grade sources is mainly an issue of energy, (3) assuming that scarcity is a major threat and (4) assuming that environmental impacts from mining and processing of mineral resources are the main problem.

The definitions above reflect a socioeconomic orientation of the evaluation of resource depletion, i.e. the notion that extraction of a resource from the natural environment leads to a decrease in its future availability for human use. This, in turn, is expressed either in relation to the available amount of a resource at a given point in time (e.g. ore deposits or fossil fuel reserves) or the future consequences (e.g. higher economic and/or energetic costs) of the extraction of a certain amount of a resource in the present. Environmental and human health impacts related to extraction or use, such as toxic emissions, are kept as separate environmental impact categories, and resource depletion directly impacting ecosystem health is not taken into account. While any transfer of a resource from the natural environment to the anthroposphere decreases availability in the natural environment at least temporarily, political and economic factors, as well as technological developments, exert their influence on the resource supply situation. A natural resource is either extracted permanently from the natural environment in the case of minerals and fossil fuels (regeneration of which is negligible on a human time scale) or subject to varying but limited regeneration rates in the case of renewable resources. The present study aims at reviewing how resources are modelled and handled in the LCIA phase of LCA and the extent to which sustainability perspectives are integrated in the assessment of metals and minerals, fossil fuels and renewable biotic resources.

We conclude, in line with previous reviews (e.g. Heijungs et al. 1997; Lindeijer et al. 2002; Finnveden et al. 2009; Hauschild et al. 2013), that there is a lack of consensus on LCIA for natural resource depletion. Marked data gaps exist in between different methods; abiotic resources, such as metals, are receiving more attention than renewables; and the various

existing methods yield incongruent results. Moreover, we find that the further from extraction impacts are modelled along the impact pathway, the less consensus there appears to be on what it is to be protected. The issue raised by Weidema et al. (2005) on whether also economic impacts, or only environmental impacts, ought to concern us in the case of resource extraction remains unresolved. In light of the recent international policy focus on resource scarcity and given that the impact in case of resource depletion is indeed on availability, this warrants closer examination in a review in order to pinpoint currently neglected areas.

The article is organised as follows: In Section 2, we provide an overview of the classification of resources in LCA; Section 3 gives an overview of current approaches to resource characterisation in LCA; Section 4 highlights the key issues related to a comprehensive assessment of resources entailing sustainability concerns; and in Section 5, we discuss the suitability of existing methods, providing an outlook for further development.

2 Natural resources classification in LCA

Natural resources are generally categorized in the context of LCA and beyond as abiotic and biotic resources or stock, fund and flow resources. Lindeijer et al. (2002) give a comprehensive set of definitions, outlined below.

Abiotic resources are inorganic or non-living materials at the moment of extraction (e.g. water, metals, also dead organic matter such as peat, coal; cf. UNEP 2010).

Biotic resources are living at least until the moment of extraction from the natural environment (e.g. wood, fish). The category does not include biotic resources reproduced by an industrial production process, as opposed to being extracted from the natural environment (e.g. livestock, fish from aquaculture, agricultural crops; wood from a plantation; cf. Guinée and Heijungs 1995; UNEP 2010).

Other approaches opt for a classification into stock, fund and flow resources, putting more emphasis on resources' capability of renewal or regrowth.

Stock resources exist as a finite, fixed amount in the natural environment, with no possibility of regrowth (e.g. rock, metals), or renewal rates on timescales too large compared to the human rate of consumption (e.g. oil).

Fund resources can be depleted at a rate dependent on a ratio of extraction to regrowth, or renewal rate. Both permanent depletion (e.g. the extinction of a resource species) and an expansion of the fund (if renewal rates exceed extraction rates) are possible.

Flow resources are resource types which cannot be depleted, although there might be local or temporal non-availability (e.g. surface freshwater dependent on a certain amount of precipitation, solar or wind energy). Renewability of flow resources is practically instantaneous.

Figure 1 gives an overview of the types of resource depletion looked at in the LCA context. No methodology provides full coverage of the resources in Fig. 1 yet; this is a general shortcoming in present methods.

Resource types are categorized here as abiotic or biotic, renewable or non-renewable and stock, fund or flow type resources.

Land use has been kept as its own category, since it is neither as clearly to be characterised in mass or volumetric terms, nor as abiotic or biotic (e.g. Goedkoop et al. 2009) but in terms of area. There is also no consensus on an unambiguous categorization of land in a stock/fund/flow schematic (Lindeijer et al. 2002; UNEP 2010), and soil is not counted as a resource as such by any methods.

Water use has been characterised along the lines of process and cooling water (Kemna et al. 2005), or fossil and standing or flowing surface water (e.g. Lindeijer et al. 2002). Its different uses place water apart from other abiotic resource types to some extent; water use is generally assessed separate from other resources (cf. Lindeijer et al. 2002; Bayart et al. 2010; Berger and Finkbeiner 2010; UNEP 2010; Giljum et al. 2011).

In this review, we will concentrate on an analysis of approaches to assess depletion of minerals, fossil fuels and biotic resources.

3 Current approaches in LCA

We review current approaches in LCA, analysing those that were chosen for the evaluation in the context of the International Reference Life Cycle Data System (ILCD) recommendation for LCIA (Hauschild et al. 2013; EC-JRC 2010a, b, EC-JRC 2011) complemented by related, further methodological developments.

The rationale behind the choice of the methods assessed for the ILCD was to consider only those including an element that reflects scarcity, defined as decreased availability for future generations caused by resource extraction, and not only inherent properties of the abiotic or biotic resource (cf. Hauschild et al. 2013).

Midpoint as well as endpoint methods are taken into account. We use the mid-/endpoint distinction as it continues to be used in LCA also in the resource context (e.g. Goedkoop et al. 2009; EC-JRC 2011; Hauschild et al. 2013).

Table 1 gives an overview of the assessment methods and the metrics used in the models analysed. At *midpoint* level, i.e. closer to the extraction of a particular resource, indicators for abiotic and biotic resources are usually based on mass; separate categories are generally used for water and land use.

Endpoint models attempt to capture the consequences of resource extraction, apart from diminishing of stocks or deposits as the most immediate impacts. This distinction is less clear in the case of resource depletion than it is with other environmental

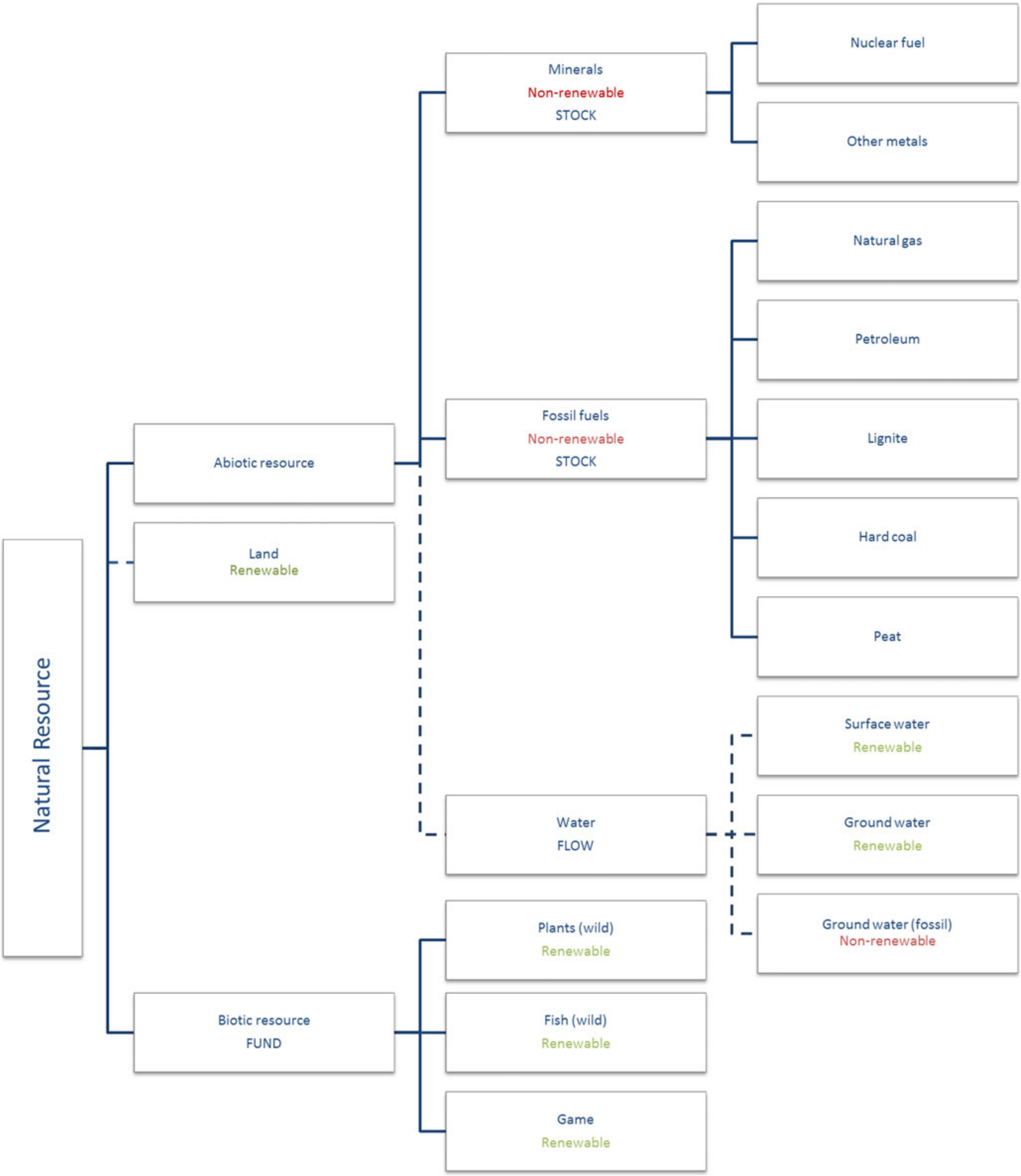

Fig. 1 A schematic of natural resources in LCA (land and water use are not covered in this review, indicated by *dashed lines*)

interventions; the impact category (resource depletion) and area of protection (natural resources) are essentially congruent.

Although water is generally defined as an abiotic resource, metrics for water use are represented in a separate column. Acknowledging the existence of different definitions of the depletion problem (Steen 2006) and expanding the differentiation, we distinguish six methods for assessing resource depletion, providing characterisation factors to be multiplied by mass of a

Table 1 Metrics for resource depletion

Model	Metric		Source
	Abiotic	Biotic	
Midpoint			
Exergy	–	–	Finnveden and Östlund (1997); Dewulf et al. (2007)
CML 2002	kg Sb-eq.; MJ		Guinée et al. (2002); van Oers et al. (2002)
EcoIndicator 99	–	–	Goedkoop and Spriensma (2001)
EcoPoints 2006	kg	kg	Frischknecht et al. (2008)
EDIP 1997	kg/cap	kg/cap	Hauschild and Wenzel (1998)
EPS 2000	kg	kg	Steen (1999)
IMPACT 2002+	kg; MJ	–	Jolliet (2008)
ReCiPe	kg Fe-eq.; MJ	–	Goedkoop et al. (2009)
Endpoint			
Exergy	MJ_{exergy}	MJ_{exergy}	Finnveden and Östlund (1997); Dewulf et al. (2007)
CML 2002	–	–	Guinée et al. (2002); van Oers et al. (2002)
EcoIndicator 99	$MJ_{surplus\ energy}$	–	Goedkoop and Spriensma (2001)
EcoPoints 2006	(Dimensionless)	(Dimensionless)	Frischknecht et al. (2008)
EDIP 1997	–	–	Hauschild and Wenzel (1998)
EPS 2000	$\$_{WTP}$	$\$_{WTP}$	Steen (1999)
IMPACT 2002+	$MJ_{surplus\ energy}$	–	Jolliet (2008)
ReCiPe	$\$_{surplus\ cost}$	–	Goedkoop et al. (2009)

Blank fields indicate the resource category is not covered

resource extracted. The underlying models are based on reserves of a resource, exergy consumption, future consequences of resource extraction (the surplus energy approach), willingness-to-pay (WTP), marginal cost of resource extraction and a distance-to-target approach.

3.1 Based on reserves and/or annual extraction rates

Indicators based on total reserves directly assess the extracted mass of a given resource, usually in relation to its deposits. Both the Institute of Environmental Sciences (CML) (Guinée and Heijungs 1995; van Oers et al. 2002) and Environmental Design of Industrial Products (EDIP) methods (Hauschild and Wenzel 1998) take this approach. These methods are currently only operational for abiotic resources, i.e. metals, minerals and fossil fuels, the extracted mass of the latter usually being converted to and expressed in energetic terms (i.e. heating value given in megajoules).

The CML method, representing the current recommendation of the ILCD (EC-JRC 2011), uses the dimensionless abiotic depletion potential (ADP), to be multiplied with the amount of a given resource extracted. The annual production of the material (the extraction rate) is divided by the reserves squared. The value for reserves is squared to take into account the fact that a simple ratio of annual production over reserve does not change if higher production rates are met by larger reserves, or low production rates by low reserves; the extraction of, e.g. 1 kg of a resource has different impacts on scarcity in these two cases.

The EDIP methodology (Hauschild and Wenzel 1998) assesses resource depletion in person reserves (given in kilograms), setting the amount extracted in relation to the respective resource's deposits given as the economic reserve (i.e. those deposits currently economically exploitable). Unlike the CML method, the EDIP approach does not reflect the current importance of a resource because it disregards extraction rates.

The CML method does include a separate impact category for biotic resources including regeneration aspects due to lack of data factors are not provided. The approach for biotic resource depletion by Heijungs et al. (1992; cf. also Lindeijer et al. 2002) uses a calculation similar to the CML method (net current extraction divided by stocks squared).

In order to overcome limitation of current methods, two other approaches were recently developed. Schneider et al (2011) propose to expand the calculation for abiotic depletion potential by including anthropogenic stocks of metals (an anthropogenic stock extended abiotic depletion potential, AADP). It is argued that inclusion of anthropogenic stocks can lead to significant changes in the representation of raw material availability; access to these stocks, however, is limited by the recyclability of any given metal. Vieira et al. (2012) propose a method which accounts for decreasing ore grades and provide an example for Cu.

3.2 Exergy

Exergy (Finnveden and Östlund 1997; Bösch et al. 2007; Dewulf et al. 2007) has been described as 'the upper limit of

the portion of a resource that can be converted into work' (Dewulf et al. 2007). Conversely, exergy extraction represents extracted potential for entropy production from the natural environment, since a resource is usually concentrated following extraction; the amount of energy necessary to bring the resource back into the state before extraction can be described as exergy loss (Lindeijer et al 2002).

The exergy method can be extended over wide range of resource types, including minerals and metals, fossil and nuclear fuel, wind, solar and hydropower, land occupation as well as atmospheric and water resources. Dewulf et al (2007), for example, calculate conversion factors (called *X factors*) in terms of exergy content per unit of the resource flow for 184 reference flows. These reference flows are taken from the ecoinvent database version 1.2 (Swiss Centre for Life-Cycle Inventories 2007). While comprehensive in scope, the approach has been criticized for possibly misrepresenting the availability of the desired quality of a non-energy resource (e.g. metals) by accounting for its exergy content (Bösch et al. 2007; Lindeijer et al. 2002). Accounting for depletion in exergy terms in a way disregards a resource's functionality and possibly limited capability of being substituted by another resource in the case of non-energy resources such as metals (Heijungs and Guinée 1997; Lindeijer et al. 2002; EC-JRC 2011; Steen 2006) and thus may misrepresent its actual scarcity.

3.3 Surplus energy

The surplus energy approach, as adopted in the Eco-Indicator 99 (EI99) (Goedkoop and Spriensma 2001) and IMPACT 2002+ (Jolliet et al. 2003), is based on the assumption that as more of a resource is extracted over time, quality of deposits still available tends to decrease. Each extraction of a certain amount of a resource from a deposit in the present will require an earlier move to more energy-intensive extraction from lower-quality, less accessible deposits in the future.

The future energy requirements for extracting a resource from lower-grade deposits are taken as an indicator for present resource depletion (cf. Mueller-Wenk 1998). The method assumes the energy requirements of current technology for extracting a given resource at a chosen point in the future (in the EI99 method, when five times the cumulative extraction up to 1990 has been mined).

A model assessing resource depletion based on the energy requirements of extraction takes declining ore grades, or less conventional fossil fuel deposits, as a premise (cf. Goedkoop and Spriensma 2001), making it not very suitable to be expanded beyond minerals and fossil fuels to include (renewable) biotic resources. This premise, of each extraction requiring an earlier move to a lower-quality, less accessible deposit in the future, does not necessarily hold for renewables.

3.4 Marginal cost

It may be argued that as energy demand increases if a resource is to be extracted from less concentrated, lower-quality deposits over time, extraction costs increase as well. A case has been made from an economic perspective for measuring resource depletion as energy demand for extraction or concentration (e.g. Roma and Pirino 2009). Monetizing the energy requirements of resource extraction, as in the ReCiPe methodology (Goedkoop et al. 2009), provides a more universally applicable indicator; in principle, marginal extraction costs can also be utilized as a metric for renewable resource extraction.

The ReCiPe 2008 method follows an idea similar to the surplus energy concept, but in addition uses monetization of surplus energy demand for characterising future efforts for resource extraction. Marginal increase of extraction cost per kilogram of extracted resource forms the basis of the model, differentiated by deposit and assuming a discount rate over an indefinite time span. Extracted amounts are converted, with iron as a reference substance (kilograms of iron-equivalent).

3.5 Willingness to pay

Willingness-to-pay (WTP) models aim to capture the monetary cost of avoiding damages to an area of protection—in our case, natural resource availability. The EPS 2000 method (Steen 1999) takes this approach to weighting of impacts from resource depletion.

A market model is used for abiotic resources, assumptions differing depending on the substance or material (different groups of metals and minerals, fossil oil, coal, natural gas): The cost of substituting a substance by a sustainable alternative is used as a WTP value for future generations affected by present-day depletion. Biotic resources or ecosystem production capacities, including fish, meat, wood and land use, are assumed to be substitutable by resources of the same kind. Market prices are used as a basis to quantify the cost of substitution.

Since the method regards individual metals to as non-substitutable and hence there is no 'sustainable alternative', the reference chosen is 1 kg of a resource as mined in the present (i.e. from present reserves). While WTP thus allows for a large, discretionary range of resource types and so allows uniform use over diverse impact categories (including land use), the assumptions and long timeframe in determining willingness to pay result in high uncertainty (EC-JRC 2011).

3.6 Distance to target

Distance-to-target approaches set environmental impacts against predefined targets. For resource depletion, such a target may be defined as a critical resource flow. The Swiss EcoPoints method (Frischknecht et al. 2008) only incorporates gravel, energy resources, land and water use. Wood, as

fuel, and uranium, as a nuclear energy carrier, are included as energy resources. The model chooses a distance-to-target approach, characterising depletion of resources in environmental load points (*Umweltbelastungspunkte*) based on the ratio of a predefined critical flow to the actual flow of a resource. In the method documentation, scarcity ratios and environmental load points are given for Switzerland only.

In principle, a ratio of critical to actual flows can be established for any natural resource or other impact categories; e.g. in the case of renewable resources, the critical flow corresponds to the carrying capacity of woods or fisheries, while in the case of non-renewable resources, such a critical flow could correspond, for example, to a defined supply horizon or a target set by policy. Non-renewable resources are multiplied by a higher factor than renewable resource flows in this method; in the case of fossil fuels, this is based here on Swiss targets for the country's energy mix and thus essentially policy-based as well.

4 Key issues for resource sustainability assessment

Several issues have so far received little attention regarding natural resource depletion within LCA impact assessment modelling. This section will examine the suitability of the models presented in Section 3 in view of these issues. The better a model covers (or can potentially cover) these aspects, the more desirable it is for future use.

Firstly, while renewal rates for biotic resources are included in the relevant models, anthropogenic stocks of, e.g. metals, and their anthropogenic stocks and recyclability, have so far not received much attention; Schneider et al. (2011) and Cummings and Seager (2008) have proposed ways to cover these aspects. We argue that both can, theoretically, be treated consistently within one methodological framework.

Secondly, resource criticality has so far received more attention outside LCA and is gaining importance in policy making (EC 2010a; NRC 2007). A resource depletion indicator reflecting the supply criticality of a given resource, subject to economic, political and strategic influences in addition to mere availability in the natural environment, is therefore desirable.

Thirdly, characterisation factors given for an individual resource can yield vastly different assessments of this resource's scarcity over the range of currently available methods. In addition, the number of resources covered varies from method to method. A user's choices of method may thus over- or deemphasize the depletion of particular resources or resource types.

4.1 Renewability of biotic resources

Renewal rates, the rates of current annual replenishment of species, are generally taken into account in the case of biotic resources, where these are covered by an LCA methodology (cf. Lindeijer et al. 2002). Nonetheless, the practical application of these methods is quite limited, and there is still a misleading perception that renewable resources are not subject to criticality in the same way as has been highlighted by recent such assessments of minerals. In fact, impacts on the carrying capacity of ecosystems and their intrinsic capability of renewal may lead to impact on human needs and life greater than shortage in, e.g. mineral resources. This issue, regarding the stock of ecological capital, is considered central in the ongoing discourse about resources in the scientific community (cf. UNEP 2007; Giljum et al. 2011; Kitzes et al. 2009; Borucke et al. 2013) but less perceived and discussed within LCA.

Renewability of resources adds a temporal element to resource depletion. Cummings and Seager (2008) give an overview of the timescale required for resource renewal. For flow resources such as wind and solar power, renewability is virtually instantaneous, while for biotic resources, renewal times range from one to several hundred years, and fossil fuels, while theoretically renewable, require geological timeframes. Metals and minerals, including nuclear fuel, as stock resources, are truly non-renewable resources (if astronomical processes are not taken into consideration). Resource renewability is less a binary question than a value on a (time) scale depending on the type of resource. Lindeijer et al (2002) propose to include biotic resources in a resource depletion assessment, subtracting current reproduction/renewal rates from current use and dividing the result by the square of the worldwide present stock of the species in question.

4.2 Recycling

In the case of metals and minerals, recycling has not been included in current models, possibly leading to misrepresentation of the availability of a substance or material (Yellishetty et al. 2009). Recycling may be regarded as a replenishment of the available anthropogenic stock of a resource, decreasing extraction rates from the natural environment. From the standpoint of resource availability from natural deposits, this is a fundamental aspect of resource efficiency. In the context of LCA, recycling implies a need for further refinement in modelling both at the inventory and at the impact assessment stages. As pointed out by Schneider et al. (2011), the anthropogenic stock of recyclable materials has to be taken into account, acknowledging the complexity of differentiating the recyclability potential of different metals.

4.3 Supply safety and critical resources

LCA approaches to resource depletion have so far focussed on the geophysical availability of a given mineral or metal, without considering the constraints of political economy, geostrategic considerations or environmental legislation in producing

countries. These latter aspects are not captured fully by existing models based, generally, on reserves and extraction rates, but may disrupt the supply of certain raw materials.

Criticality, assessed in the European policy context by the Ad-hoc Working Group on defining critical raw materials (EC 2010a), applies to several industrial minerals and metals (cf. EC 2011a, b; Schueler et al. 2011) and is presently determined by economic and geopolitical factors: economic importance, supply risk and environmental country risk (i.e. stricter environmental regulation in an exporting country impairing imports of a resource type).

The method for criticality assessment put forward by the European Commission is based on three separate indicators: First, economic importance is denoted by share in consumption and gross value-added of a resource's end-use sector. Second, the supply risk of a resource is measured taking into account the stability of the producing countries, substitutability and recycling rate. Third, the environmental country risk is established as a combination of the environmental performance index of the producing countries, substitutability of the resource in question and the recycling rate. Fourteen raw materials are listed as critical: antimony, beryllium, cobalt, fluorspar, gallium, germanium, graphite, indium, magnesium, niobium, platinum group metals, rare earth elements, tantalum and tungsten.

A similar concept has been proposed by the US National Research Council (NRC 2007). Natural resources are placed in a 'criticality matrix' showing the importance in use (corresponding to the potential impact of supply restriction) and availability (corresponding to the supply risk) of a given resource. Importance of a mineral is calculated as a weighted product of the proportion of the mineral in the US market and the impact of supply restriction; the assessment of supply risk incorporates geological, technical, regulatory, environmental, political and economic availability. In comparison to the European Commission's indicators, the US approach is less rigidly defined, depending ultimately on assessment by an expert committee.

Buijs and Sievers (2011) show in their survey of relevant indicators that the question of resource criticality is of an socioeconomic or strategic rather than environmental nature, with inherently stronger fluctuations and biased models as opposed to a regard simply towards, e.g. deposits vs. extraction rates. Criteria for identifying critical raw materials are discretionary and still under debate, and the list of critical raw materials may change accordingly. Figure 2 gives an example, reporting three indicators used for criticality assessment.

Figure 2 shows the abiotic depletion potential, as used in the CML methodology (Guinée and Heijungs 1995; van Oers et al. 2002), together with the three different indicators developed by the European Commission's Ad-hoc Working Group on defining critical raw materials (EC 2010a). CML, used as an example of a well-accepted mass-based indicator here, divides the annual production (i.e. the extraction rate) of a given resource by the ultimate reserves (the concentration of an element in the earth's crust) squared. The results use over antimony as a reference resource. Indicators for criticality are likewise shown as antimony-equivalents here.

The figure shows that there is no clear correlation between a safe resource supply, as calculated by the resource criticality indicators, and the reserve–extraction rate ratio. If an indicator for resource depletion is to measure the availability of a natural resource, the parameters above hold useful additional information. So far, this aspect has been virtually absent from LCA. A thorough analysis of the topic is provided by Mancini et al. (2013), as a result of an extensive discussion on criticality and LCA by different stakeholders, from research, government and industry. Besides, critical/non-critical status may apply to biotic just as it does for abiotic resources, reflecting danger to renewability (if the ecological carrying capacity of the system is exceeded) and/or risk of species' extinction. Including only those biotic resources that are under pressure from extraction (e.g. Lindeijer et al. 2002) can serve as a starting point for a representation of criticality. It is suggested to include scarce biological resources if they are listed either an important global resource (fish or forest resources) by the Food and Agriculture Organization of the United Nations, or listed as vulnerable, endangered or critically endangered (corresponding to defined net extraction to stock ratios) by the International Union for the Conservation of Nature. Extraction and renewal rates are proposed to be estimated over 10 years, or three generations of the species, if that is a longer period than 10 years (cf. Section 4.1). The critical status of species reported by the IUCN corresponds roughly to a scale of current-net-extraction-to-stock ratios, with <0.2 representing the lowest and >0.8 the highest risk to the species (Lindeijer et al 2002).

Critical status of species has been defined before as comprising the importance of and threat to a species (De Groot et al. 2003). Narrowing a multi-faceted definition of importance down to importance as a global resource, as in Lindeijer et al (2002), better meets the definition of 'natural resource' as it is used in the LCA context (i.e. resources for human use).

4.4 Gaps and differences in characterisation

Figure 3 shows a comparison of characterisation factors for a selected set of minerals and crude oil as a fossil energy source from the models outlined in Section 4. The characterisation factors have been normalized over iron as one of the most abundant metals (cf. EC 2010b). The EcoPoints 2006 methodology (Frischknecht et al. 2008) is not included in the figure due to its small number of resources covered (especially minerals) limiting its comparability in this case.

While resource depletion metrics may differ (cf. Table 1), Fig. 3 highlights the variations of the degree of scarcity of

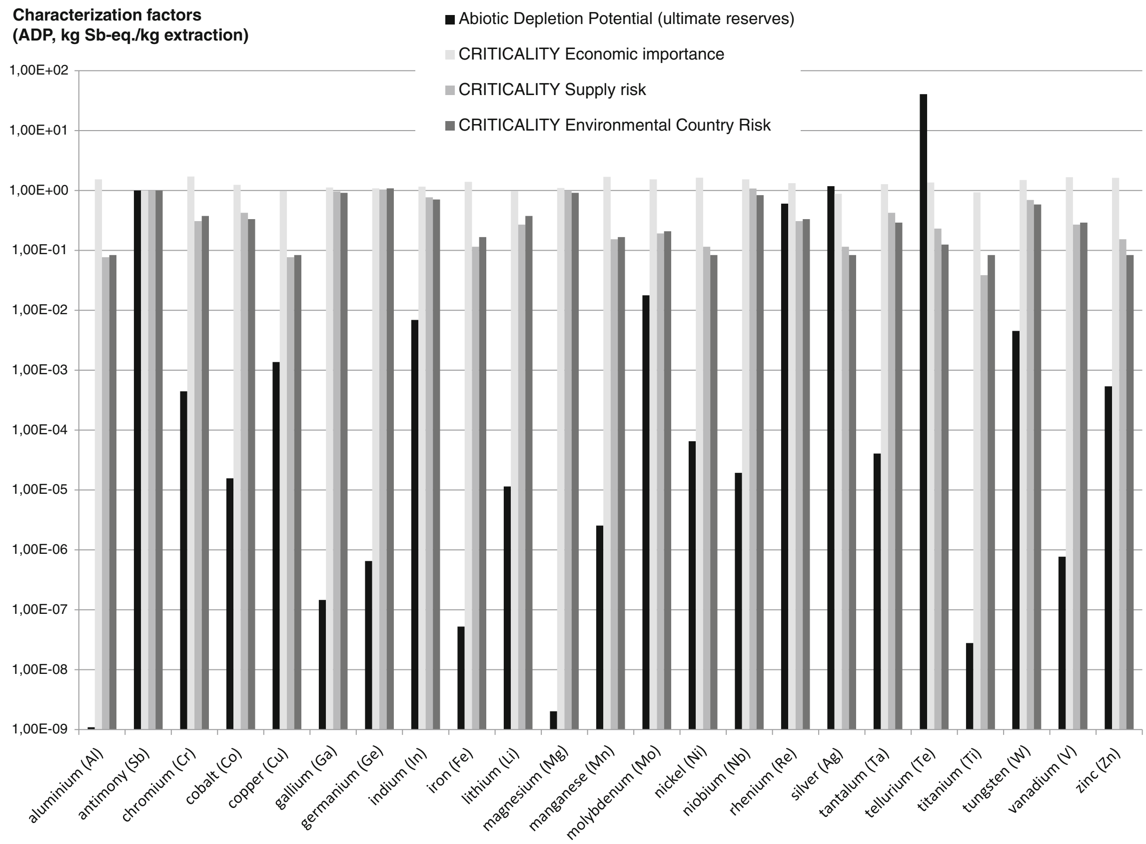

Fig. 2 Mass-based indicator (abiotic depletion potential; van Oers et al. 2002, cf. Electronic Supplementary Material) and critical resource indicators according to EC (2010a, b)

individual resources relative to each other, as shown by different methods. Owing to the abundance of iron on earth, characterisation factors for the metals selected here indicate, with the exception of aluminium in the CML, EPS and ReCiPe methods, higher scarcity than iron.

Characterisation factors for individual substances differ by several orders of magnitude. The comparison shows, moreover, that a similar theoretical basis of impact assessment methods does not correspond to a similar ranking of resources with respect to scarcity/depletion potential. For example, the EDIP 97 and CML 2002 methodologies are mass-based, yet show considerable discrepancies in mineral depletion indicators in relation to iron.

One resource's relative importance is thus strongly dependent on the model chosen by the user. A comprehensive and unbiased assessment would therefore necessitate the impracticable task of looking at the currently available range of methods covering partly congruent sets of resources in parallel.

Coverage of resources is similarly inconsistent as characterisation; Table 2 compares the number of resources for which current methods give characterisation factors. Fossil fuels include crude oil, natural gas, brown coal, hard coal, peat and sulphur. Uranium is counted as a nuclear fuel, separate from other minerals, to highlight the fact that it is not included in every method. Biotic resources comprise wood and wild animals (including fish).

Similar to the issue of varying characterisation, the choice of method influences the outcome with regard to resource depletion as a result of the number of resources included in the impact assessment. A high number of resources covered are evidently more desirable to render a complete picture than a more limited scope.

5 Discussion

There is a notable lack of consensus across impact assessment methods for natural resource depletion in LCA. This review shows that there are several loose ends in related but not yet combined impact assessment methods for natural resource depletion. These offer insight into which steps can be taken towards a more unified approach that takes into account the areas for improvement outlined.

Current methods tend to cover only limited, and often different, sets of resource types and aspects to resource depletion, presenting a somewhat fragmented picture. An indicator that is comprehensive, scientifically robust, but parsimonious and easy to understand is desirable.

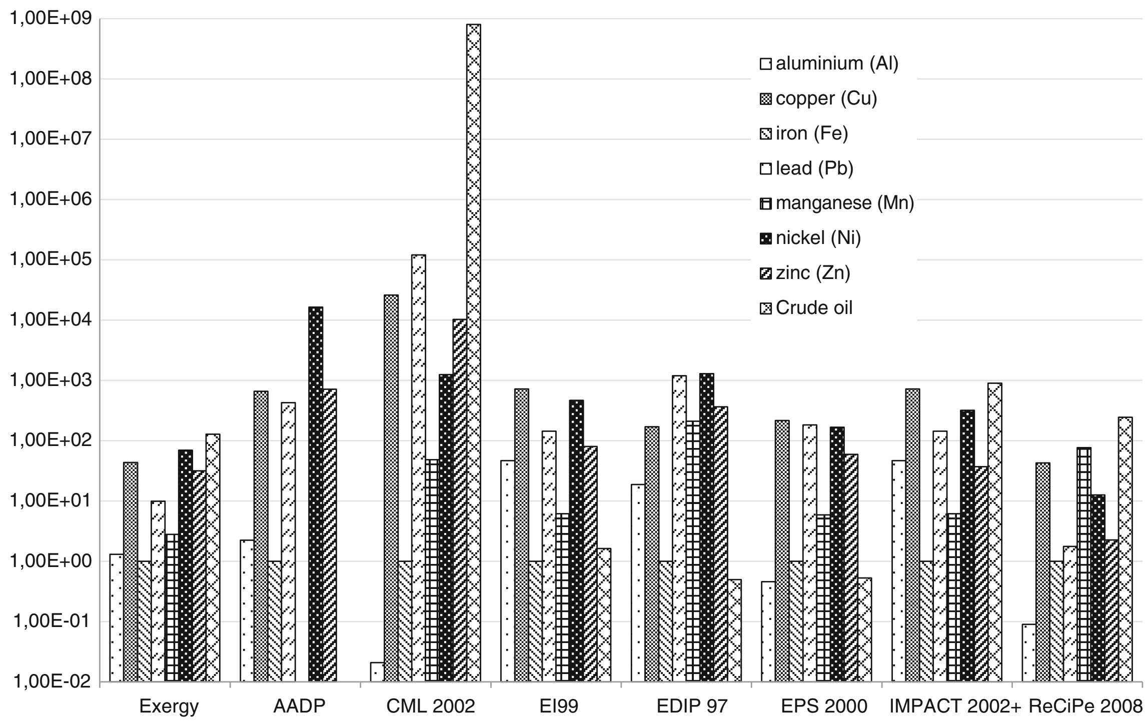

Fig. 3 Characterisation (at midpoint) of selected resources, normalized over iron (cf. Electronic Supplementary Material). No characterisation is given by Schneider et al. (2011) for manganese and crude oil using the AADP method

We identify three main issues which offer room for improvement. Firstly, if biotic resources are covered by resource depletion models, natural renewal rates have to be taken into account; the effect of recycling on the supply of non-renewable resources such as metals, however, has so far received little regard in LCA impact assessment models and may be prone to double counting and misrepresenting resource availability. Secondly, scarcity in the natural environment is looked at by present models, whereas different levels of criticality and supply safety, or threat of disruption of raw material flows influenced by socio-economic factors, are not assessed within these same frameworks. Thirdly, current approaches are disparate in their methodologies and the types and number of resources covered.

Following a systematic overview of resource types in LCA and methodological approaches, the issues above are examined in relation to current models in order to highlight present shortcomings in these areas. The results of the analysis inform our outlook on the inherent opportunities of current methods for further development, incorporating a greater range of resource types and aspects to the depletion issue in a meaningful way.

At midpoint, closer to the environmental intervention (extraction), a combination of separate approaches is possible by adjusting or expanding existing mass-based indicators. Section 4.4 highlights the need for expanding coverage of existing methods over a greater number of resources.

Several proposals have been made that take the CML method (Guinée and Heijungs 1995; van Oers et al 2002) as a point of departure. The basic calculation method has been modified by Lindeijer et al. (2002), Schneider et al. (2011) and

Table 2 Number of natural resources covered per method (cf. Electronic Supplementary Material); where different sources for a resource (i.e. ore grades or energy contents) are given, the resource in question is only counted once

	Exergy (Dewulf et al. 2007)	CML 2002 (van Oers et al. 2002)	EI 99 (Goedkoop and Spriensma 2001)	EcoPoints 2006 (Frischknecht et al. 2008)	EDIP 97 (Hauschild and Wenzel 1998)	EPS 2000 (Steen 1999)	IMPACT 2002+ (Jolliet et al. 2003)	ReCiPe 2008 (Goedkoop et al. 2009)
Abiotic: minerals	57	48	12	1	29	67	13	19
Abiotic: fossil fuels	6	4	4	4	4	3	5	4
Abiotic: nuclear fuel	1	1		1	1	1	1	1
Biotic	5			1		2	1	

AADP is not listed here, since Schneider et al. (2011) only give factors for ten materials as examples

Berger and Finkbeiner (2010) to cover biotic resources, freshwater depletion and anthropogenic stocks, respectively. For biotic resources (as for freshwater depletion), the renewal rate of the resource in question is introduced by simply subtracting it from a corresponding extraction rate.

The issue of incorporating renewability of biotic resources has, partially, been resolved through this approach by proposing a straightforward net-extraction-rate-over-stocks calculation. Further adding the anthropogenic stocks of a resource to the amount present in natural stocks or deposits makes it possible to draw a more realistic picture of raw material availability. To avoid double counting, a more explicit introduction of recycling does not seem advisable; an inclusion into deposits or stocks of a resource of anthropogenic stocks gives a picture of recycling rates (as a replenishment of anthropogenic stocks). This has been done in the form of the AADP, the adaptation in Schneider et al (2011) of the calculation method for ADP. This is makes sense, however, only under the assumption that anthropogenic stocks are at least as accessible as the corresponding resource currently extracted from nature, e.g. the quality of the considered anthropogenic stocks of a metal would have to be able to be substituted for the same amount extracted from natural ores.

More far-reaching, socioeconomic considerations of resource criticality and supply safety (see Section 4.3) are not to be meaningfully covered by an approach so close to the immediate environmental intervention at hand, i.e. the extraction process diminishing biophysical resource availability. The recently published indicators on critical raw materials (EC 2010a; NRC 2007; UNEP 2010) are so far limited to metals. The proposal of Lindeijer et al (2002) to include only biotic resources under a certain and defined threat of extinction into a metric of biotic resource depletion would incorporate the criticality aspect only at the cost of giving an incomplete picture of resource use and would result in an even less complete assessment if extended to abiotic resources.

The exergy method (Finnveden and Östlund 1997; cf. also Dewulf et al. 2007) has the advantage of wide applicability over various resource types and stability over time in characterisation factors (being based on thermodynamic properties); there is no consensus, however, as to whether an indicator for exergy use would depict diminishing stocks (or scarcity) of, e.g. a mineral (Heijungs and Guinée 1997; EC-JRC 2011). A mass/volume-based indicator, taking into account available deposits, has shown to be quite adaptable to reach better coverage and more accurate assessments of scarcity, as has been shown by the adaptations of the CML method mentioned above.

At endpoint, the aim is to cover the impacts of reduced availability of a resource beyond the simple reduction in stocks and deposits. This can be achieved by assessing the additional effort necessary for extracting a given amount of a resource in the future. Surplus energy approaches, as well as willingness-to-pay models, provide metrics that are, in theory, applicable to a wide range of resources.

Exergy, however, keeps to depicting biophysical resource availability. The ReCiPe method (Goedkoop et al. 2009) introduces the marginal extraction cost per extraction of functional unit instead of energy, thereby using a more accessible metric. By monetizing increased extraction efforts, it is theoretically possible to include a comprehensive range of resources using one metric, without making changes to this methodological framework.

In the case of minerals, the notion of decreasing ore grades and related increased technological extraction efforts being directly caused by the exhaustion of more accessible deposits, and leading to increased prices as an indicator for mineral depletion, has been criticized elsewhere (Mudd and Ward 2008; West 2011). Mudd and Ward show technological progress, and not necessarily exhaustion of high-grade ores, as a driver for extraction from lower-grade deposits at largely stable prices. Monetization of the extraction effort per functional unit provides a more complete picture of supply constraints if one is to measure more far-reaching impacts of extracting a certain amount of a given resource. West (2011) argues that political and socioeconomic constraints borne out of environmental policy considerations (e.g. carbon pricing, regulation of toxic emissions from mining activities), as opposed to mere biophysical availability, will eventually prove a limiting factor for resource availability. A marginal extraction cost model, as opposed to one based on energetic, chemical/physical terms such as exergy or surplus energy, may therefore turn out to draw a more useful picture. Although Weidema et al. (2005) suggest to model future technologies (human activity) in the inventory analysis, but at the same time acknowledge the need for socioeconomic impact assessment of resource use and since West (2011) notes environmental policy as a driver of extraction cost, it can be argued that marginal cost can capture the environmental and socioeconomic impacts of resource extraction—environmental impacts, in this case, by proxy of policy-induced surplus cost. Any inclusion of economic and social impacts, however, does muddle the distinction of human activity (causing impacts) and the natural environment (impacted upon) raised by Weidema et al (2005). Conversely, forgoing the modelling of future scenarios for resource extraction would leave only midpoint indicators such as exergy extraction or extraction-rate/reserve-based characterisation models as valid options (cf. also Finnveden et al. 2009).

Zhang et al. (2010) propose a differentiated, hierarchical Eco-LCA approach to account for ecosystem services as ecological resources, exceeding the relatively narrow definition of availability of a natural resource for human use considered by the other models in this review. From a set of individual resource flows, aggregated midpoint indicators are established using mass, exergy or energy metrics. These can be further used to establish endpoint indicators. While only renewability and

resource efficiency are included in the example given by Zhang et al. (2010), they consider a wider range of endpoint indicators possible, such as monetary valuation or an ecological footprint. This does not, however, overcome the limitations we mentioned inherent in mass- or energy-consumption-based indicators used here.

While the indicators for resource criticality referred to in Section 4.3 attempt to put discrete numbers on a variety of socioeconomic factors influencing raw material supply, this does not seem practicable in a parsimonious and easy-to-understand indicator as it is understood in this review. As our example showed, criticality indicators do not correspond to models focussed on resource depletion from the natural environment. While the indicators may not be easily integrated in an existing framework, they can yield complementary information on a resource's actual availability.

This article identifies readily available room for improvement for resource depletion indicators in an LCA context. Such possibilities are more evident at the midpoint level intended to give a more immediate picture of biophysical depletion. With regard to anthropogenic stocks, renewability of biotic and recycling of abiotic resources, present approaches can be combined without basic methodological changes, to give a better indication of depletion, not least by expanding the scope at least of mass-based indicators over a greater number of resources. At endpoint, the less immediate, societal or economic impacts are, arguably, at present best covered by indicators quantified in terms of economic costs such as marginal extraction cost of mineral resources; monetization of impacts clearly presents room for expansion of these present models.

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References

Bayart JB, Bulle C, Deschenes L, Margni M, Pfister S, Vince F, Koehler A (2010) A framework for assessing off-stream freshwater use in LCA. Int J Life Cycle Assess 15(5):439–453

Berger M, Finkbeiner M (2010) Water footprinting: how to address water use in life cycle assessment? Sustainability 2010(2):919–944

Borucke M, Moore D, Cranston G, Gracey K, Iha K, Larson J, Lazarous E, Morales JC, Wackernagel M, Galli A (2013) Accounting for demand and supply of the biosphere's regenerative capacity: the National Footprint Accounts' underlying methodology and framework. Ecol Indicators 24:518–533

Bösch ME, Hellweg S, Huijbregts MAJ, Frischknecht R (2007) Applying cumulative exergy demand (CExD) indicators to the ecoinvent database. Int J Life Cycle Assess 12(3):181–190

Buijs B, Sievers H (2011) Critical thinking about critical minerals, Clingendael International Energy Program (CIEP) and Bundesanstalt für Geowissenschaften und Rohstoffe (BGR), Clingendael Briefing Paper. http://www.europesworld.org/NewEnglish/Home_old/CommunityPosts/tabid/809/PostID/2800/CriticalThinkingaboutCriticalMinerals.aspx. Accessed Jul 2012

Cummings C, Seager T (2008) Estimating exergy renewability for sustainability assessment of corn ethanol. IEE International Symposium on Electronics and the Environment, May 2008, San Francisco

De Groot R, van der Perk J, Chiesura A, van Vliet A (2003) Importance and threat as determining factors for criticality of natural capital. Ecol Econ 44(2–3):187–204

Dewulf J, Boesch ME, de Meester B, van der Vorst G, van Langenhove H, Hellweg S, Huijbregts AJ (2007) Cumulative Exergy Extraction from the Natural Environment (CEENE): a comprehensive life cycle impact assessment method for resource accounting. Environ Sci Technol 41(24):8477–8483

EC (European Commission) (2008) "The raw materials initiative—meeting our critical needs for growth and jobs in Europe". COM (2008) 699 Communication European Commission, Brussels

EC (2010a) Critical raw materials for the EU: report of the Ad-hoc Working Group on defining critical raw materials. European Commission DG Enterprise, Brussels

EC (2010b) Annex V to the report of the Ad-hoc Working Group on defining critical raw materials. European Commission DG Enterprise, Brussels

EC (2011a) A resource-efficient Europe—Flagship initiative under the Europe 2020 Strategy. European Commission, Brussels

EC (2011b) Tackling the challenges in commodity markets and on raw materials. COM (2011) 25 Communication. European Commission, Brussels

EC-JRC (European Commission Joint Research Centre) (2010a) ILCD handbook. Analysis of existing environmental impact assessment methodologies for use in life cycle assessment, 1st edn. IES, Joint Research Centre, Ispra. Available at http://lct.jrc.ec.europa.eu

EC-JRC (2010b) ILCD handbook. Framework and requirements for life cycle assessment models and indicators, 1st edn. IES, Joint Research Centre, Ispra. Available at http://lct.jrc.ec.europa.eu

EC-JRC (2011) ILCD handbook. Recommendations based on existing environmental impact assessment models and factors for life cycle assessment in European context. IES, Joint Research Centre, Ispra. Available at http://lct.jrc.ec.europa.eu

EEA (European Environment Agency) (2010) The European environment—state and outlook 2010. Part 4: natural resources and waste. EEA, Copenhagen

Eurostat (2013) Eurostat's economy-wide material flow accounts. http://epp.eurostat.ec.europa.eu/statistics_explained/index.php/Material_flow_accounts. Accessed Jan 2013

Finnveden G, Östlund P (1997) Exergies of natural resources in life-cycle assessment and other applications. Energy 22(9):923–931

Finnveden G, Hauschild MZ, Ekvall T, Guinée J, Heijungs R, Hellweg S, Koehler A, Pennington D, Suh S (2009) Recent developments in life cycle assessment. J Environ Manag 91:1–21

Frischknecht R, Steiner R, Jungbluth N (2008) Oekobilanzen—Methode der oekologischen Knappheit—Oekofaktoren 2006. BAFU, Zurich

Giljum S, Burger E, Hinterberger F, Lutter S, Bruckner M (2011) A comprehensive set of resource use indicators from the micro to the macro level. Resour Conserv Recycl 55(3):300–308

Goedkoop M, Heijungs R, Huijbregts M, de Schryver A, Struijs J, van Zelm R (2009) ReCiPe 2008. A life cycle assessment method which comprises harmonized category indicators at the midpoint and the endpoint level. Report I: characterisation. Ministry of Housing, Spatial Planning and Environment, Amsterdam

Goedkoop M, Spriensma R (2001) The Eco-indicator 99: a damage-oriented method for life cycle impact assessment—methodology report. PRe Consultants, Amersfoort

Guinée J, Heijungs R (1995) A proposal for the definition of resource equivalency factors for use in product life-cycle assessment. Environ Toxicol Chem 14(5):917–925

Guinée JB, Gorrée M, Heijungs R, Huppes G, Kleijn R, van Oers L, Wegener Sleeswijk A, Suh S, Udo de Haes HA, de Bruijn H, van

Duin R, Huijbregts MAJ (2002) Life Cycle Assessment: An Operational Guide to the ISO Standards. Kluwer Academic Publishers, Dordrecht

Hauschild M, Wenzel H (1998) Environmental assessment of products, vol 2—scientific background. Chapman & Hall, UK

Hauschild MZ, Goedkoop M, Guinée J, Heijungs R, Huijbregts M, Jolliet O, Margni M, de Schryver A, Humbert S, Laurent A, Sala S, Pant R (2013) Identifying best existing practice for characterization modelling in life cycle impact assessment. Int J Life Cycle Assess 18(3):683–697

Heijungs R, Guinée JB, Huppes G, Lamkreijer RM, Udo de Haes, HA, Wegener Sleeswijk A, Ansems AMM, Eggels PG, van Duin R, de Goede HP (1992) Environmental life cycle assessment of products. Guide (part 1) and background (part 2). CML Leiden University, Leiden

Heijungs R, Guinée J, Huppes G (1997) Impact categories for natural resources and land use: survey and analysis of existing and proposed methods in the context of environmental life cycle assessment. CML Leiden University, Leiden

Jolliet O, Margni M, Charles R, Humbert S, Payet J, Rebitzer G, Rosenbaum R (2003) IMPACT 2002+: a new life cycle impact assessment methodology. Int J Life Cycle Assess 8(6):324–330

Kemna R, van Elburg M, Li W, van Holstein R (2005) MEEuP—the methodology report. European Commission, Brussels

Kitzes J, Galli A, Bagliani M, Barrett J, Dige G, Ede S, Erb K, Giljum S, Haberl H, Jungwirth S, Lenzen M, Lewis K, Loh J, Marchettini N, Messinger H, Milne K, Moles R, Monfreda C, Moran D, Nakano K, Pyhala A, Rees W, Simmons C, Wackernagel M, Wada Y, Walsh C, Wiedmann T (2009) A research agenda for improving national Ecological Footprint accounts. Ecol Econ 68:1991–2007

Lindeijer E, Mueller-Wenk R, Steen B (2002) Impact assessment of resources and land use. In: Udo de Haes HA et al (eds) Life-cycle impact assessment: striving towards best practice. SETAC, Pensacola

Mancini L, De Camillis C, Pennington D (2013) Security of supply and scarcity of raw materials: a methodological framework for sustainability assessment. European Commission, Joint Research Centre, Institute for Environment and Sustainability, Publications Office of the European Union, Luxemburg

Mudd GM, Ward JD (2008) Will sustainability constraints cause 'peak minerals'? Paper presented at the 3rd International Conference on Sustainability Engineering & Science: Blueprints for Sustainable Infrastructure, Auckland, New Zealand, 9–12 Dec 2008

Mueller-Wenk R (1998) Depletion of abiotic resources weighted on base of "virtual" impacts of lower grade deposits used in future. IWOe Universitaet St. Gallen, St. Gallen

NRC (National Research Council) (2007) Minerals, critical minerals, and the US economy. National Academies, Washington, DC

Roma A, Pirino D (2009) The extraction of natural resources: the role of thermodynamic efficiency. Ecol Econ 68(10):2594–2606

Sala S, Farioli F, Zamagni A (2012) Progress in sustainability science: lessons learnt from current methodologies for sustainability assessment. Part 1. Int J Life Cycle Assess. doi:10.1007/s11367-012-0508-6

Schneider L, Berger M, Finkbeiner M (2011) The anthropogenic stock extended abiotic depletion potential (AADP) as new parameterization to model the depletion of resources. Int J Life Cycle Assess 16(9):929–936

Schueler D, Buchert M, Liu R, Dittrich S, Merz C (2011) Study on rare earths and their recycling. Oeko-Institut e.V, Freiburg

Steen BA (1999) A systematic approach to environmental priority strategies in product life development (EPS). Version 2000—models and data of the default method. Chalmers University, Lindholmen

Steen BA (2006) Abiotic resource depletion different perceptions of the problem with mineral deposits. Int J Life Cycle Assess 11(1):49–54

Swiss Centre for Life-Cycle Inventories (2007) Ecoinvent database. Swiss Centre for Life-Cycle Inventories, Duebendorf. www.ecoinvent.org. Accessed May 2012

UNEP (2007) Global environmental outlook 4. United Nations Environment Programme, Nairobi

UNEP (2010) Assessing the environmental impacts of consumption and production: priority products and materials. UNEP International Panel for Resource Management, Paris

van der Voet E, van Oers L, de Bruyn S, de Jong F, Tukker A (2009) CML report 184. Environmental impact of the use of natural resources and products. CML Leiden University, Leiden

van Oers L, de Koning A, Guinée JB, Huppes G (2002) Abiotic resource depletion in LCA. Road and Hydraulic Engineering Institute, Ministry of Transport and Water, Amsterdam

Vieira MD, Goedkoop MJ, Storm P, Huijbregts MAJ (2012) Ore grade decrease as life cycle impact indicator for metal scarcity: the case of copper. Environ Sci Technol 46(23):12772–12778

Weidema BP, Finnveden G, Stewart M (2005) Impacts from resource use—a common position paper. International Int J Life Cycle Assess 10(6):382–382

West J (2011) Decreasing metal ore grades: are they really being driven by the depletion of high-grade deposits? J Ind Ecol 15(2):165–168

Yellishetty M, Ranjith PG, Tharumarajah A, Bhosale S (2009) Life cycle assessment in the minerals and metals sector: a critical review of selected issues and challenges. Int J Life Cycle Assess 14(3):257–267

Zhang Y, Baral A, Bakshi BR (2010) Accounting for ecosystem services in life cycle assessment, part II: toward an ecologically based LCA. Environ Sci Technol 44(7):2624–2631